



Calculating the benefit of conservation actions

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Abstract

The benefit (or additionality) attributable to a conservation action is the difference between the outcomes of two scenarios: (1) the scenario with the conservation action, and (2) the alternative scenario, in which action did not occur. However, many conservation decisions are made using approaches that do not appropriately calculate this benefit. We review recent scientific literature and conservation policies to examine how conservation benefit is calculated in three situations: systematic reserve selection, investment in agri-environment schemes, and biodiversity offset trades. In the examples we considered, the approaches used to calculate conservation benefit often involved assumptions about the alternative scenario that were not explicit, demonstrably wrong or both. We suggest that assumptions about how conservation value changes over time in the alternative scenario can often be substantially refined, and that making these assumptions explicit by calculating directly the expected difference between the two scenarios is likely to improve the quality of conservation decision-making.

Introduction

Sound decisions about land purchase for reservation, investment in conservation actions on private land, or ensuring a fair exchange in a biodiversity offset scenario all rely on a transparent and logical accounting framework (Murdoch *et al.* 2007; BBOP 2009; Gibbons *et al.* 2009; Hajkowicz *et al.* 2009). Simply put, all conservation decisions require an estimate of the payoff, or conservation benefit, attributable to the action. This benefit, V_{benefit} in Figure 1, is the difference between the outcomes of two scenarios: (1) the scenario in which the conservation action occurs (represented by the bold line in Figure 1), and (2) the scenario in which it does not (the “alternative scenario”; also referred to as the baseline or counterfactual, represented by the faint line) (Ferraro and

Pattanayak 2006; BBOP 2009; EPA 2010; Gibbons 2010; Gordon *et al.* 2011). The difference between the outcomes of the two scenarios is often referred to as “additionality” (McKenney & Kiesecker 2010).

Achieving cost-effective outcomes for conservation—the greatest gain for a given cost or the least cost for a given gain—is essential given the universal constraint of limited resources (Shogren *et al.* 1999; Murdoch *et al.* 2010). A sound approach to calculating expected benefit underpins innumerable conservation decisions, such as through allowing evaluation of the cost-effectiveness of alternative policy options (Joseph *et al.* 2009; Khanna & Ando 2009), comparison of competing bids in a reverse auction for provision of ecological goods (e.g., Stoneham *et al.* 2003), or estimation of equivalence in a biodiversity offset trade (e.g., Gordon *et al.* 2011). An

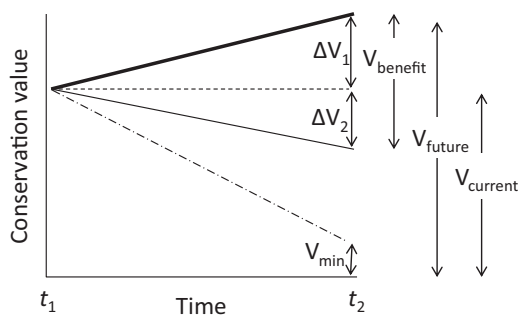


Figure 1 Change in conservation value (V) at a site between two points in time (t_1 and t_2) with (bold line) and without (faint line) implementation of a conservation action. The change in the conservation value attributable to the conservation action at time t_2 is V_{benefit} (the difference between the two scenarios), but the conservation benefit of the action is often calculated as V_{future} or V_{current} . In this example, the conservation action results in both an increase in conservation value (e.g., due to reduced pest density) compared to that at t_1 (ΔV_1) and averted loss of value (ΔV_2). V_{min} represents a future minimum value that is legally possible within duty of care (see Table 1). Strictly, the conservation benefit should be calculated based not only for the difference between two scenarios at a particular point in time (t_2 in Fig. 1), but also for how distant that time horizon is (and ideally the nature of the pathways toward t_2) (Moilanen *et al.* 2009).

approach that does not account explicitly for both the with-action and the alternative scenarios, as well as associated costs, can therefore lead to suboptimal decisions and limited additionality (Ferraro & Pattanayak 2006; Hodge & Reader 2010). Although the importance of explicit estimation of additionality based on comparison with a clear alternative scenario is recognized in the literature (e.g., Ferraro & Pattanayak 2006; Wunder *et al.* 2008; Joseph *et al.* 2009; Hodge and Reader 2010; Murdoch *et al.* 2010), in practice, many conservation decisions are made using approaches that do not appropriately calculate conservation benefit.

Here, we review approaches used in conservation policy and planning to calculate conservation benefits of actions. We examine how benefit is calculated, regardless of the specific conservation objective, in order to make decisions for three important conservation approaches: systematic reserve selection, allocation of agri-environment expenditure, and biodiversity offsetting. We conclude that a flaw shared by many of the approaches used to calculate conservation benefit is that the assumptions about the alternative scenario are not explicit, demonstrably wrong or both. We contend that information often exists that can substantially refine assumptions about change in conservation value over time in the alternative scenario, and that even when such information is limited, making these assumptions explicit and transparent is likely to improve the quality and transparency of conservation decision-making.

Calculation of conservation benefit

Calculating the expected benefit of a conservation action—such as the purchase of a new reserve—requires (1) explicit estimation of the change in conservation value (e.g., population size of a threatened species) both with, and without, the action taking place; and (2) calculation of the difference between these two scenarios (V_{benefit} in Figure 1). Thus, not only is it important to quantify how conservation value might change with investment in a conservation action over some time period, but also how that value would be expected to change in the absence of the action over the same period. For example, the purchase of land to add to a conservation network implies that: (1) some threat is present that may cause a decline in conservation value, and (2) the purchase is expected to reduce this threat. Therefore, to calculate the expected benefit of the purchase, explicit consideration of the threat to the site or network's conservation value both with and without the action is required. Failure to do this potentially leads to poor conservation decisions and missed opportunity to achieve maximum conservation benefit (Pressey *et al.* 2004; Strange *et al.* 2006; Wilson *et al.* 2006; Merenlender *et al.* 2009).

A failure to specify explicitly the alternative scenario simply means that implicit assumptions about that scenario are made. If those assumptions do not hold, then poor outcomes may result. For example, conservation benefit from the protection of a site may erroneously be equated with the site's current conservation value (V_{current} in Figure 1a). This involves the implicit assumptions that conservation value of the site is zero if (for example) we do not protect it as a reserve, and that if we do protect it, its conservation value will not change. In other words, threat is assumed to be absolute in the alternative scenario, and fully abated if protected. Unless these assumptions hold, we will have incorrectly estimated the benefit of that conservation action, potentially leading to poor decisions.

Of course, the alternative scenario cannot be known with certainty. Nevertheless, threat to the conservation value of a site is rarely absolute or absent, and tends to vary systematically with factors such as patch area, human population density, landform and soil, and degree of legislative protection (McKenney 2001; Pressey & Taffs 2001; Burgess *et al.* 2006; Maron & Fitzsimons 2007; Duncan & Dorrough 2009; Maron *et al.* 2010). Yet knowledge about variation in threat is not routinely incorporated into conservation decision-making. Similarly, some beneficial land management actions may be likely to continue regardless of the payment of an incentive, whereas others are costly and unlikely to occur without an incentive payment (Wunder *et al.* 2008; Hodge and Reader

2010). Importantly, the conservation benefit from an action comprises not only the benefit at a target site, but also benefits that the action may have outside of the site (Moilanen 2008). Estimation of both the “with action” and the alternative scenario is challenging, but it cannot be avoided—all prioritization incorporates assumptions about both scenarios, even if these assumptions are implicit. Therefore, even without investing in collecting more information to improve the accuracy of scenarios, making the implicit assumptions explicit is likely to improve the effectiveness of decision-making.

Despite recognition in the scientific literature of the importance of a logical and realistic approach to calculation of benefit from conservation actions (e.g., Wilson *et al.* 2005), translation of this principle into conservation practice often lags behind. In the following three sections we review approaches used in conservation planning and policy to calculate conservation benefit, in the context of: (1) systematic reserve selection, (2) agri-environment schemes, and (3) biodiversity offsets.

Systematic reserve selection

Over the past few decades, systematic approaches to reserve selection have become important tools for conservation planning (Watson *et al.* 2011). One of the primary advantages of a systematic approach is that it is explicit about conservation objectives and assumptions, which allows a formal prioritization of actions using decision theoretic approaches (Sarkar *et al.* 2006; Wilson *et al.* 2009). The principal benefit from the gazetted of reserves is the removal or reduction of threats, and so incorporating threat into estimating the benefit from investment in a reserve is essential for informing investment decisions. Yet, despite considerable interest in dealing with and incorporating threats into systematic conservation planning (Meir *et al.* 2004; Wilson *et al.* 2006; Moilanen & Cabeza 2007; Pressey *et al.* 2007), threat is still ignored in many examples of systematic reserve selection in the literature (e.g., Kremen *et al.* 2008; Fuller *et al.* 2010).

To assess the extent to which threat is explicitly incorporated into systematic reserve selection, we searched the recent peer-reviewed literature for studies on reserve selection and the grey literature for conservation plans that identify spatial priorities for the protection of biodiversity (see Supporting Information for methodology). From this literature we examined how 71 studies and six plans accounted for threat in the metrics of conservation benefit used to evaluate contributions of reserves toward conservation objectives.

From the peer-reviewed literature, only 14% of studies (10 studies) explicitly incorporated threats in any form

into their metrics of conservation benefit (although a further 11% [eight studies] implicitly considered threats in an ad hoc way by incorporating adjustments to cost, species' weightings, or conservation targets). The remaining studies implicitly equated benefit with current site value (V_{current} in Figure 1). Of the ten studies that did explicitly consider threat, six only considered threat in the “reserved” or conservation action scenario (thus calculating benefit as ΔV_2), three only considered threat in the conservation action scenario (equating benefit with V_{future}), and only one built threat into both. Each of the six conservation plans we reviewed from the grey literature (Supporting Information) considered threat only in the alternative scenario and assumed no change in value if conservation were to occur, thus estimating benefit as ΔV_2 . However, even though data on threat in the alternative scenario were considered, they were not used explicitly to calculate conservation benefit. Instead, threat was either incorporated using multicriteria evaluation with current conservation value (V_{current}) and threat as criteria (*sensu* Noss *et al.* 2002; Pressey *et al.* 2004; Pressey & Bottrill 2008), or in an ad hoc way through adjustments to targets or costs.

An assumption commonly made was that conservation action removes all threats (i.e., $V_{\text{current}} = V_{\text{future}}$). Reservation is often an effective means of preventing habitat loss, but may be less effective at ameliorating threats such as inappropriate fire or invasive species (Howes & Maron 2009). In addition, although the way most conservation plans incorporate threat in the alternative scenario using multicriteria evaluation has been shown to work reasonably well (Pressey *et al.* 2004), the necessarily subjective choice of relative weightings applied to conservation value and threat are likely to influence priorities considerably. When data on conservation value and threat are both available, a more logical use of these data would be to calculate explicitly the conservation benefit (V_{benefit}).

Systematic reserve selection has focused predominantly on achieving representation of conservation features in reserve systems, rather than the retention, or persistence, of conservation features within the region of interest (Pressey *et al.* 2004). For example, in Australia, the National Reserve System has explicit targets for the representation of biodiversity within reserves (National Reserve System Task Group 2009), but has no specific targets for the retention of biodiversity in Australia as a whole, which is a quite different but more relevant objective. Our review of the literature reflects this issue, with assumptions about threat often implicit or incorporated in an ad hoc way, with the result that metrics used for prioritization often do not reflect the true benefit of conservation actions.

The single example that we found in our review that explicitly accounted for threat in the reserved and alternative scenarios (Game *et al.* 2008) used a simple model of the health of reefs in the Great Barrier Reef under both scenarios to illustrate how to prioritize marine protected areas. This example shows that a clear specification of the conservation objective and reformulation of the metric of conservation value can address the issues we identified, and the process can be usefully informed by existing data and simple, but plausible and explicit, assumptions about changes in biodiversity under the conservation action and alternative scenarios (e.g., White *et al.* 2012).

Agri-environment schemes

There is considerable global expenditure on agri-environment schemes, which typically involve public investment in the conservation of biodiversity on private farmland (Hajkowicz *et al.* 2009). The increasing popularity of market-based instruments for the delivery of such funding has led to a proliferation of benefit metrics for comparison of competing bids for funding (Collins & Scoccimarro 2008), particularly in Australia. These metrics, or utility functions, are derived through a variety of approaches and are intended to represent the benefit being purchased for a given investment in private land conservation, often in order to compare the cost-effectiveness of competing investment options (Stoneham *et al.* 2003). We examined recent (since 2000) agri-environment schemes that used a quantitative metric of benefit to compare competing bids for funding (see Supporting Information for methodology) and converted the approaches used to derive the benefit metric into a common notation to allow comparison, regardless of the type of conservation outcomes targeted by the scheme (Table 1).

We found that rather than calculating conservation benefit as the difference between the “with investment” scenario and the alternative scenario, the difference between the current value and the estimated future value of a site (with investment) was often used (ΔV_1 in Figure 1; issue category A in Table 1). In effect, this reflects an assumption that the alternative scenario for a site is one of no change from its current state. Related literature on payment for ecosystem services (PES) schemes suggests a similar approach is commonly used; of 14 PES schemes reviewed by Wunder *et al.* (2008), only five were explicit about the alternative (or baseline) scenario, and four of those five used a static baseline representing the current value of provision of services.

In most cases, the estimated change in conservation value was then weighted by a function of the current

conservation value of the site to derive the benefit metric (issue category B in Table 1). This means that good opportunities for cost-effective gains through improving the condition of poor-quality sites, for example, are more likely to be missed. In some cases, marginal gains in conservation value can be greater in poorer-quality sites (Huth & Possingham 2011). Scaling benefit by current conservation value implies that a unit of benefit at a site that is in poor condition is not the same as a unit of benefit in a site of high condition. Such a nonlinear scale is more challenging to interpret and translate, and we suggest a linear scale for units of benefit would be far more intuitive and transparent. Alternatively, at least in cases where current site value was an additive factor in the metric, its relative weighting may reflect an implicit assumption about the alternative scenario—the probability the site’s value would decline to zero in the absence of investment.

In one case—that of the Environmental Stewardship Scheme for Box-Gum Grassy Woodland (Gibbons & Ryan 2008)—benefit was calculated by using a static minimum value possible for the site within statutory “duty of care” as the alternative scenario, and taking the difference between this (represented by V_{\min} in Figure 1) and the estimated future value with investment at t_2 (V_{future} in Figure 1). This procedure reflects the conceptual approach that we propose, as it involves calculating the difference between two future scenarios. However, it means that the alternative scenario was derived based on the “worst-case” (legally permissible) scenario for the site, rather than the most probable scenario in the absence of the investment. This approach is used to avoid penalizing landholders whose land management performance is already above duty of care, but it introduces a bias toward overestimation of benefit from investment in higher-quality sites.

Biodiversity offsets

Biodiversity offsets often use approaches similar to those of agri-environment schemes both to calculate the quantum of biodiversity lost at a development site in order to identify how much must be generated elsewhere to achieve at least “no net loss” of biodiversity, and to calculate the benefit from actions at the offset site (Gibbons *et al.* 2009). Careful calculation of such gains at offset sites is essential to ensure they are genuinely additional, as well as equivalent with the lost biodiversity values at the development site (Gibbons & Lindenmayer 2007; Maron *et al.* 2010; Quétier & Lavorel 2011; Maron *et al.* 2012). However, standard approaches for explicit quantification of benefit are still relatively rare. Because

Table 1 Examples of approaches for calculating conservation benefit of investment in private land conservation incentive schemes (“agri-environment schemes”) and for use in offset loss-gain calculations. Issue categories: A = benefit is entirely or partly the difference between current value (or current value adjusted for “duty of care”) and future value; B = addition or multiplication by current site value; C = benefit is assumed to be equivalent to current or future value. In most cases, benefits are per unit area, and may be weighted by the duration and security of commitment to management or the conservation priority of the target values. Here, we limit our attention to the component of the calculation which attempts to capture benefit per unit area

Region	Plan/Scheme	Benefit metric	Issue	Reference
<i>Agri-environment schemes</i>				
USA	Conservation Reserve Program (Wildlife Factor)	$(V_{\text{future}} \text{ or } V_{\text{current}}) + \Delta V_1^*$	A, B	USDA 2011
England	Countryside Stewardship	$V_{\text{current}} + \Delta V_1$	A, B	Short <i>et al.</i> 2000
Lower Saxony, Germany	Market-based payment scheme for plant diversity	V_{current}	C	Ulber <i>et al.</i> 2010
Fitzroy Basin, Australia	Fitzroy Basin Association Biodiversity Tender	$V_{\text{current}} + \Delta V_1$	A, B	Windle & Rolfe 2006
Queensland	Nature Assist	$V_{\text{current}} + V_{\text{benefit}}$	B	Hajkowicz <i>et al.</i> 2008
Australia	WWF Auction for Landscape Recovery	$V_{\text{current}} \times \Delta V_1$	A, B	Gole <i>et al.</i> 2005
Victoria, Australia	Bush Tender	$V_{\text{current}} \times V_{\text{benefit}}$	B	Stoneham <i>et al.</i> 2003; DSE 2008
Desert Uplands, Australia	Desert Uplands Landscape Linkages Auction	V_{current}	C	Windle <i>et al.</i> 2009
New South Wales, Australia	Environmental Services Scheme	$V_{\text{current}} \times \Delta V_1$	A, B	Oliver <i>et al.</i> 2005
South Australia, Australia	Mt Lofty Ranges conservation auction	$V_{\text{current}} \times \Delta V_1$	A, B	Crossman <i>et al.</i> 2011
Australia	Environmental Stewardship scheme for Box Gum Grassy Woodland 2008–2010	$V_{\text{future}} - V_{\text{min}}^\dagger$	A	Gibbons & Ryan 2008
<i>Offset schemes</i>				
England	Proposed offsets scheme	ΔV_1	A	DEFRA 2011
Australia	EPBC Act Environmental Offsets Policy	V_{benefit}	–	DSEWPC 2012
Queensland, Australia	Biodiversity Offset Policy	V_{current}	C	DERM 2011
Victoria, Australia	Vegetation Gain Approach used for native vegetation offsets	$(V_{\text{future}} - V_{\text{min}}) + (V_{\text{current}}/10^\ddagger)$	B	DSE 2006
New South Wales, Australia	BioBanking Methodology	$(V_{\text{min}}/10) + \Delta V_1$	A, B	DECC 2008

V_{current} = current value of a site; ΔV_1 = estimated increase in value with management/investment; ΔV_2 = estimated averted loss of value with management/investment; V_{future} = estimated future value of a site with management/investment; V_{benefit} = difference between the estimated future value of a site with and without management/investment; V_{min} = minimum (legally) possible future value of a site within statutory duty of care (current site value, adjusted for all possible permitted impacts). Often, scores against multiple criteria are combined to give “value”; for a single metric, V_{current} may be composed of different criteria than ΔV_1 or ΔV_2 .

* Either existing or planned vegetation cover suitable for wildlife could be used to generate equivalent “points.”

[†]Scheme considered for inclusion only sites beyond a certain value of V_{current} .

[‡]When statutory protection of a site is increased, an additional percentage of current site score is added to the gain estimate.

few offset schemes are detailed in the peer-reviewed literature, we searched for information on current and draft or proposed offset schemes listed in two recent reviews (BBOP 2009; Madsen *et al.* 2010) as well as conducting a web search for the term “offset policy.”

For most schemes, information on the approach used to estimate conservation benefit (or “credits”) at an offset site was unavailable, offsets were indirect (such as cash payments), or no systematic approach for calculating conservation benefit at offset sites existed (e.g., Ezemvelo KZN Wildlife 2009). For example, the well-established species conservation banking approach in the United States is administered by individual banks, which do not necessarily follow federal or state policy guidance (Fox & Nino-Murcia 2005). Credits in species conservation banking are generally allocated per acre of habitat enrolled (Bauer *et al.* 2004; Fox & Nino-Murcia 2005). This may be interpreted as equating ben-

efit with the current value of a site (issue category A), but the information available suggested that the approach varied substantially among banks and the detail of credit calculation was generally unavailable. For only five approaches were we able to clearly identify the approach used for calculation of benefit at offset sites, and these approaches were converted to a common notation for comparison, as for the agri-environment metrics (Table 1).

Offset schemes commonly allow credits to be generated through “averted loss,” or the protection of habitat which otherwise may have been under threat (ΔV_2 in Figure 1; e.g., DSE 2006; DERM 2011; DSEWPC 2012). The amount of benefit, or credit, from an offset action is therefore highly dependent on how threat is incorporated into the construction of alternative scenarios (Gordon *et al.* 2011). Yet in four of the five examples we reviewed, the calculation of benefit

attributed to the protection of existing vegetation was not scaled appropriately to threat. For example, although the Australian state of Victoria's policy on scoring gain does directly calculate the benefit from ongoing and improved site management, "averted loss" benefit attributable to increased legal protection of the site is calculated simply as a fixed percentage of V_{current} (Table 1; DSE 2006). The value of V_{current} increases with higher-quality, more intact vegetation, and with patch size (as it is scored according to the Habitat Hectares approach; Parkes *et al.* 2003). Thus, there is an implicit assumption that the threat to the site is directly proportional to site quality. However, higher-quality sites are afforded greater legislative protection (DSE 2002), which suggests an inverse relationship between threat and quality may be a more realistic assumption.

In addition to variation in levels of legislative protection, threat to the persistence of vegetation tends to vary systematically with soil fertility, land cost and vegetation condition (Pressey *et al.* 1996; Merenlender *et al.* 2009), yet biodiversity benefit metrics and mitigation credit calculators rarely account for this. In some cases, risk that an area may come under pressure for development is considered to reduce the potential benefit from protection. For example, Kiesecker *et al.* (2009) in developing an offset prioritization approach for a suite of biodiversity assets affected by a Wyoming gasfield development, explicitly blocked selection of areas with a high potential for future oil and gas development. Yet given the high "averted loss" potential of such areas, it is possible that some may have been incorporated in an optimal solution (depending on the strength of the available protection).

The Australian Environment Protection and Biodiversity Conservation Act environmental offsets policy (DSEWPC 2012) was the only policy we located that involved explicit and direct measurement of the estimated difference between the two scenarios. The approach also was the only one that required a time horizon by which the benefit was expected to accrue to be stated explicitly, and employed time discounting to allow fair comparison between future benefit (at the offset site) and immediate loss (at the impact site) (Moilanen *et al.* 2009).

Conclusion

Considerable investment and activity in biodiversity conservation is translated into on-ground actions based on decisions driven by conservation benefit metrics. Whether these decisions involve prioritizing direct investment in reserves or agri-environment schemes,

or are used to balance loss with gain in biodiversity offsetting, the objective is to identify—and often maximize for a given expenditure—the conservation benefit. Conservation benefit is simply the difference between the future conservation value with the action and the future value without the action. Although the importance of calculating benefit, or additionality, in this way is widely (but not universally) acknowledged in the literature (e.g., Ferraro & Pattanayak 2006; Murdoch *et al.* 2010), we have shown here that most approaches used to support decision-making deviate from this approach. Most approaches failed to appropriately estimate benefit because they (a) ignored or made unrealistic assumptions about the "with action" or the alternative scenarios, or both, and/or (b) weighted estimated benefit by the current conservation value of the site (through addition or multiplication). This means that priorities for biodiversity conservation are often driven by current biodiversity values, rather than the potential conservation benefit of a conservation action.

The consequences of incorrect calculations of expected benefit are pervasive and potentially serious. For example, in a biodiversity offset scenario, assumptions about the baseline or alternative scenario directly determine how many credits are allocated for a conservation action (Gordon *et al.* 2011), which in turn at least partially determines allowable losses elsewhere in jurisdictions with a stated goal of no net loss. For example, protecting a large, high-quality site may seem favorable, but if little threat is abated and this fact is not accounted for, then the benefit of the protection (and generated offset credits) will be overestimated. Relative cost-effectiveness of alternative conservation actions is sensitive to assumptions about future scenarios and the approach used to estimate benefit; thus, inappropriate assumptions and approaches result in less-beneficial conservation networks or other outcomes. This means not only that limited resources are being used less efficiently, but also that this inefficiency may not be recognized.

We suggest modeling of realistic alternative scenarios that explicitly incorporate threat is an important step in conservation decision-making, allowing comparison among alternative futures to be more explicit. Construction of both the "with action" and alternative scenarios is often difficult because of uncertainty in both ecological trajectories and policy environments. However, even with relatively large uncertainties, building explicit and realistic scenarios by, for example, harnessing expert opinion can improve the conservation prioritization process (e.g., see Joseph *et al.* 2009; Visconti *et al.* 2010; Cawardine *et al.* 2012). Further, all approaches that attempt to calculate the benefit attributable to a decision have built-in assumptions about the alternative

scenario—the assumptions are simply often implicit, ignored or avoided. Ensuring the approach used for calculating conservation benefit is appropriate does not guarantee “correct” answers—this will depend on the accuracy with which the scenarios are estimated. However, making explicit the assumptions and uncertainty associated with estimating scenarios should encourage ongoing improvement in how conservation decisions are made.

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Supporting Information

Additional Supporting Information may be found in the online version of this article at the publisher's web site:

Methodology for literature searches

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